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A management framework for Nunavik beluga

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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ABSTRACT

The signing of land claim agreements with Canada's Inuit represented historic moments in Canada's history. These agreements established co-management responsibilities for wildlife resources between the Inuit and Government of Canada. The co-management of wildlife resources transferred basic wildlife management responsibilities to co-management boards and restricted the veto powers of the responsible minister to overturn Board decisions. This has resulted in an apparent management paradox, since on one hand the Government's ability to limit harvesting has been restricted, while on the other hand, Canada has international responsibilities to ensure a management structure based on Maximum Sustainable Yield (MSY) and the Precautionary Approach (PA). The rights of hunters to harvest are often highlighted in discussions with hunters, but other aspects of the land claim agreements call for the development of management systems that also respects the principles of conservation and continued sustainability of the resource. Thus, it would appear that the development of a PA/MSY based framework is consistent with the concepts and principles of recent land claim agreements implemented in Canada. Different management frameworks, some of which have been developed to manage marine mammal stocks in Canada were examined using the eastern Hudson Bay beluga as a study case. Other stocks were also examined for discussion. Criteria were developed that could be used to determine if a stock could be managed using a framework where the probability of reaching the management objective within a specified timeframe can be identified explicitly, or whether a more general approach referred to as the Potential Biological Removal (PBR) should be used to set Total Allowable Harvest levels. Criteria were also developed that could be used to determine an appropriate Recovery Factor (F_R) that could be used as part of the PBR calculation. Examples presented here are for illustration only.

Un cadre de gestion pour le béluga du Nunavik

RÉSUMÉ

La signature d'accords sur des revendications territoriales avec les Inuits du Canada a été marquante dans l'histoire canadienne. Ces accords ont établi les responsabilités de cogestion des ressources fauniques incombant aux Inuits et au gouvernement du Canada. En vertu du principe de cogestion, les responsabilités de base relatives à la gestion des ressources fauniques ont été transférées à des conseils de cogestion, et le pouvoir d'infirmer les décisions des conseils associé au droit de veto du ministre responsable a été restreint. Cela a donné lieu à un paradoxe apparent en matière de gestion. En effet, d'une part, la capacité du gouvernement à limiter l'exploitation a été restreinte, mais d'autre part, le Canada est responsable devant la communauté internationale d'assurer une structure de gestion fondée sur le rendement maximal soutenu (RMS) et l'approche de précaution (AP). Les droits de récolte des chasseurs sont souvent évoqués au cours des discussions avec les chasseurs, mais d'autres aspects des accords sur des revendications territoriales requièrent l'élaboration de systèmes de gestion respectant également les principes de conservation et de durabilité de la ressource. Par conséquent, il semble que l'élaboration d'un cadre de travail fondé sur l'approche de précaution et le rendement maximal soutenu soit conforme aux concepts et aux principes des récents accords sur des revendications territoriales mises en œuvre au Canada. Divers cadres de gestion, dont certains ayant été élaborés pour gérer les stocks de mammifères marins au Canada, ont été examinés en utilisant les bélugas de l'Est de la baie d'Hudson comme cas d'espèce. D'autres stocks ont également été examinés à des fins de discussion. On a élaboré des critères pouvant être utilisés en vue de déterminer si un stock peut être géré à l'aide d'un cadre de travail lorsque la probabilité d'atteindre l'objectif de gestion peut être déterminée explicitement dans les délais prescrits, ou si une approche plus générale connue sous le nom de retrait biologique potentiel (RBP) devrait être utilisée pour établir les niveaux du total autorisé des captures. On a également établi des critères pour déterminer un facteur de récupération (F_R) approprié pouvant être utilisé dans le cadre du calcul du RBP. Les exemples fournis dans le présent document ne sont présentés qu'à des fins d'illustration.

INTRODUCTION

Two key aspects of the precautionary approach (PA) are that decisions are to be more cautious when information is less certain, and decision rules for stock management when the resource reaches clearly stated reference points are defined in advance (Punt and Smith 2001). This represents a fundamental shift in philosophy away from one of post-damage control (possibly via civil liability) towards a system of anticipatory or pre-damage control of risks (UNESCO 2005). The underlying concept of applying the PA for a conservation objective was outlined in the United Nations Convention on the Law of the Sea (UNCLOS), which calls on signatories to ensure 'through proper conservation and management measures that the maintenance of the living resources in the exclusive economic zone is not endangered by over-exploitation. Such measures shall also be designed to maintain or restore populations of harvested species at levels which can produce the maximum sustainable yield'. As a signatory to the United Nations Convention on the Law of the Sea, Canada further entrenched its commitment to PA and adopted a Sustainable Fisheries Framework (DFO 2006) which provides guidelines for managing resources at or above the level capable of producing Maximum Sustainable Yield (MSY).

The objective of any management approach is to manage the population at, or above, a Precautionary Reference level, (PRL, also referred to as the Upper Stock Limit) (Fig. 1). A second limit, referred to as a Limit Reference Level (LRL), signifying a level below which significant harm can occur, is also established. This creates three zones, Healthy, Cautious and Critical. Any population above the PRL is considered to be in the Healthy Zone while a population below the LRL is considered to be in the Critical Zone. The space between the LRL and the PRL provides a buffer zone, referred to as the Cautious Zone. It should be set at an appropriate distance above the LRL to provide sufficient opportunity for the management system to recognize a declining stock status and sufficient time for management actions to have effect. The Target Reference Level (TRL), the level to which management and industry would like to see the population, can be identified at, or above, the PRL. For a stock falling below the PRL, harvesting must progressively be reduced to avoid reaching the LRL.

In practice, many fisheries management agencies identify the PRL and LRL based on the population size that is at, or above, a level that is capable of producing MSY. This differs from the original use of MSY, where the objective was to achieve harvests equivalent to, or slightly below, MSY. While MSY is, in principle, relatively simple and well understood, it is a theoretical construct and difficult to quantify; the dynamic nature of MSY and MSY reference points may not be sufficiently addressed by current frameworks, hence the absolute maximum MSY may virtually never be obtainable (Sissenwine et al. 2014). Despite these challenges, the specification of target and limit reference levels, and associated harvest control rules, has generally resulted in a reduction in the number of stocks that are overfished or fisheries where overfishing is occurring (*ibid*).

While PA (and MSY) frameworks may have adopted different forms, most have identified conditions for populations falling into one of two categories depending on how well we understand the population dynamics. These are often referred to as 'Data Poor' and 'Data Rich'. In the case of Data Poor stocks, our understanding of stock status is highly uncertain and therefore, harvest advice is more risk adverse to avoid causing significant harm to the resource. In the case of Data Rich stocks, however, knowledge concerning the stock is higher, and a higher level of risk may be accepted depending on whether the stock is considered to be in a 'Healthy', 'Cautious' or 'Critical' zone. For a 'Healthy' stock, harvest decisions may be made for reasons other than conservation, whereas conservation becomes an increasingly important reason for limiting harvesting for stocks in the Cautious or Critical zones (e.g. DFO 2006;

Hammill and Stenson 2007; Stenson et al. 2012; DFO 2013). This general framework has been applied to the management of seals in Atlantic Canada since 2003 and is similar to those used in other jurisdictions (e.g. see ICES 2012; Minister of Fisheries 2008).

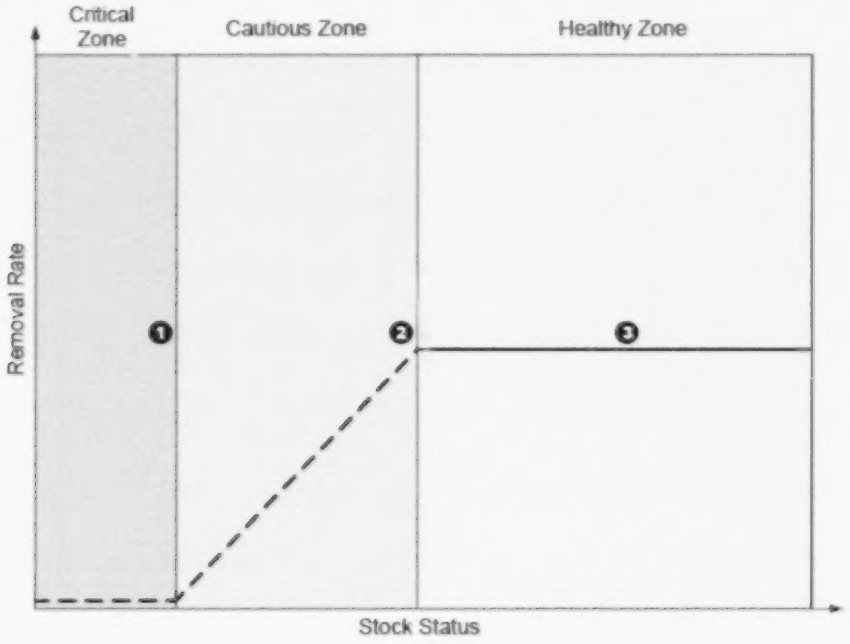

Figure 1. The generalized PA framework for fisheries. 1. The Limit reference level (LRL). 2. The precautionary reference level (PRL). 3. A removal rate identified to maintain the resource within the Healthy zone.

The distinction between Data Poor and Data Rich is not always clear and there may be a gradient in our understanding of dynamics running from stocks for which we know very little to stocks where we have considerable quantitative information. Instead of the simple Data Rich and Data Poor dichotomy, ICES has identified six categories, where a category 1 stock has a full quantitative assessment, while Categories 5 and 6 are data-poor stocks where data are very limited, in some cases including stocks for which only landings data are available.

The signing of land claim agreements with Canada's Inuit represented historic moments in Canada's history. These agreements recognized Inuit rights, ownership over certain areas and established co-management responsibilities for wildlife resources between the Inuit and Government of Canada. The co-management of wildlife resources transferred basic wildlife management responsibilities to co-management boards and although the ultimate responsibility for wildlife management remains with the Government, the Government agrees to exercise this responsibility in accordance with the provisions of the agreement which means that the veto powers of the responsible minister over harvesting practices are severely restricted.

This has created an apparent management paradox. Under the land-claim agreements, the ability of the Minister to limit subsistence harvesting has been severely restricted. At the same time, to be consistent with domestic policy and as a signatory of international agreements, Canada has an international responsibility to ensure the implementation of PA/ MSY management frameworks and the sustainable use of its marine resources (e.g. United Nations

Convention on the Law of the Sea (UNCLOS) and United Nations Conference on Straddling Fish Stocks and Highly Migratory Fish Stocks). This means identifying management objectives and specific catch limits to provide a means of evaluating management compliance and effectiveness. However, development of clearly defined PA frameworks to manage northern marine mammal harvests has not occurred to date. Some recent events suggest that wildlife management in northern Canada will be increasingly driven by international pressure, with the threat of loss or increasingly complicated access to international markets unless Canada alters its management framework. At the same time, the international attempt to pose "urban European'" values on the use of wildlife has resulted in increasing conflict with respect to co-management system (e.g. see Suluk and Blakney 2008; Lovecraft and Meek 2011; Weber et al. 2015; Dale and Armitage 2011) and complicates efforts to develop PA frameworks.

The Nunavik Inuit Land Claim Agreement (NILCA) established the Nunavik Marine Region Management Board (NMRWB) as the main instrument of wildlife management in the Nunavik Marine Region (section 5.2 of NILCA) and the Board sets total allowable take levels for all species within the Region (section 5.3 of NILCA). Decisions by NMRWB, or a Minister, to restrict Nunavik Inuit harvesting may only do so to the extent necessary for one of three reasons:

1) To effect a conservation purpose in accordance with the agreement
2) For the purposes of allocation
3) To provide for public health or safety.

The land claim recognizes certain principles: that the human population is increasing; a long-term, healthy, renewable resource economy is both viable and desirable; there is a need for an effective system of wildlife management (i.e. management framework) that respects both Inuit harvesting rights and priorities, and provides optimum protection to the renewable resource economy; and the wildlife management system and the exercise of Nunavik Inuit harvesting rights must follow the principles of conservation. The principles of conservation are defined as: the maintenance of the natural balance of ecological systems; the maintenance of vital, healthy wildlife populations capable of sustaining harvesting needs; the protection of wildlife habitat; and the restoration and revitalization of depleted populations of wildlife and wildlife habitat.

To date much of the discussion has revolved around the issue of unreasonable restrictions of hunter's rights to harvest. However, the land claim is also concerned with the need for conservation, the development of a management framework, and continued sustainability. Thus it would appear that the development of a PA/ MSY based framework is consistent with the concepts and principles of recent land claim agreements implemented in Canada.

The development of PA (or a management framework linked with MSY) is an iterative process involving scientists, hunters and managers. Therefore, what is presented here is not a definitive framework, but an example of approaches that could be used to manage EHB beluga. The main objective is to find a balance between the limitations to unduly restricting subsistence harvesting (Section 5.5.3), the obligations to respect the principles of conservation (sections 5.1.4-5.1.5), and the need for a management framework that balances protecting harvesting rights and priorities with providing optimum protection to the renewable resource economy (section 5.1.2) as well as to respect DFO policy and our international obligations.

PA has been discussed frequently by NMMPRC in the past, as well as elsewhere within the Department (DFO 2006; Hammill and Stenson 2007, 2009, 2013; Stenson et al. 2012; DFO 2013). The concepts of Data Rich and Data Poor, have been accepted and developed within the context of commercial harvesting of seals, and PRL and LRL have been discussed and/or identified (Hammill and Stenson 2003, 2007, 2009, 2013; Stenson et al. 2012). For harp, hood

and grey seals, Data Rich is defined as having three or more abundance estimates over a 15-year period, with the last estimate obtained within the last five years, and current information (within the last five years) on fecundity and/or mortality to determine sustainable levels of exploitation (Stenson et al. 2012). If these data are available, then harvest advice is provided as the probability that different catch levels will respect specific management objectives. If little is known about the stock, then harvest recommendations have been provided using the Potential Biological Removal (PBR) method. It must be noted that the Data Rich/Poor criteria were developed for seals where a time series of demographic parameters has been available, often for a decade or more. Within the context of subsistence harvesting for cetaceans such data are usually not available. Harvest advice following different approaches has been provided for a number of Arctic cetaceans, but there has been limited discussion about the application or development of PA and the criteria required. In the past, the committee has provided harvest advice using the PBR approach estimated from aerial survey estimates of abundance (e.g. Eastern Canada-Western Greenland (EC-WG) Bowhead), a PBR estimated from a population model of abundance (Cumberland Sound (CS) beluga) and using a probabilistic approach, where the risk of a population decline over 10 years at different levels of harvest is identified using a population model (Eastern Hudson Bay (EHB) beluga). These approaches have been applied based upon qualitative evaluations of the information available on the stock rather than more specific criteria that can be applied consistently across populations. Consequently, there is a need for appropriate criteria that will allow stakeholders, scientists and managers to evaluate the approach needed for providing management advice.

The DFO guidelines suggest that for fish populations, the LRL be set at 40% of the population size at MSY, and the PRL be set at 80% of the population size at MSY. For a population below the LRL, i.e. in the critical zone, managers should limit harvesting to an absolute minimum; the aim is to developing rebuilding plans that have a high probability ($\geq$75%), of the population moving into the cautious zone within 1.5 to 2 generations. Once in the cautious zone, the management response varies but should aim for the stock to increase with a high probability and move into the Healthy zone within 1.5-2 generations.

The Atlantic Seal Management (ASM) framework provides an alternative approach to identifying the reference levels and has been applied to management of seal harvesting in Atlantic Canada. Under the ASM framework, the LRL and PRL were established as proportions of the largest population observed or estimated (N_{max}) (Hammill and Stenson 2003, 2007; Stenson et al. 2012) rather than a theoretical or assumed K. The management objectives are to set harvests where there is an 80% probability that the population will remain above the PRL. The PRL is set at 70% of N_{max}, while the LRL is set at 30% of N_{max}. This framework is consistent with the DFO framework in that the PRL and LRL delineate healthy, cautious and critical zones. It is also consistent with approaches used by other international organizations managing marine mammals such as the International Whaling Commission and the Norwegian/Russian Sealing Commission.

Under the ASM, total removals have been estimated using the PBR approach when information on abundance or dynamics of the stock are limited, (i.e. data poor). PBR was developed by the United States for the management of marine mammals under the Marine Mammal Protection Act (MMPA). The management objective of PBR is to manage the population above the Optimum Sustainable Population (OSP) level which is the population size similar to that capable of providing MSY. Simulations that included a variety of uncertainties have shown that removals at PBR levels have a 95% probability that the population will increase above, or remain above, OSP over a period of 100 years. The strength of the PBR approach is that it only requires a single abundance estimate to calculate a PA compliant removal level. However, when several estimates are available it is not clear which estimate is to be used. Furthermore, with the

possible exception of altering the Recovery Factor (Fr), the use of PBR does not consider the status of a population; PBR treats a stock that is quite small and has only a single estimate of abundance the same as one which is very abundant (i.e. well above OSP) with several available abundance estimates.

Other approaches have been applied to marine mammal management in Canada. The current management objective for EHB beluga is referred to as Sustainable Yield (SY), which identifies the catch that maintains a constant population over a period of time. In the case of EHB beluga, it is the catch that has a 50% probability of the population not declining over a period of 10 years. SY is not considered PA compliant because it does not take into account the status of the population, does not allow for any population recovery and does not establish any buffer in case of unexpected events (e.g. see McLaren et al. 2001). For some marine mammal stocks a recovery target has been identified, with a time frame for recovery and a target population size set at 70% of the pre-commercial biomass (B_0 although for marine mammals - N_0 is used instead of B_0) (DFO 2005). In the case of EHB beluga, N_0 was identified as 12,500, resulting in a Recovery target of 8,750 animals (DFO 2005). In a more recent analysis, N_0 was estimated assuming that Struck and Loss (i.e. whales killed but not recovered or reported) levels were very low. This resulted in an estimate of N_0=8,000, which would mean a recovery target of 5,600 belugas (Doniol-Valcroze et al. 2012). The Recovery framework is directed to meet requirements of the Species at Risk Act, but this framework is not PA compliant.

Ecosystems and Fisheries Management has requested that a PA framework be developed that could be used in the management of EHB beluga to set the Total Allowable Take (TAT). Here we explore options that might be used to develop a PA framework for the management of Nunavik beluga and possibly for other cetaceans harvested in northern Canada. We begin with a set of criteria that together could be used to evaluate our level of understanding of abundance and dynamics of a stock (i.e., data rich or data poor). Then we investigate the impact of which management framework is used to provide harvest advice for the stock (e.g., MSY, PBR, etc.). We begin with the case of EHB beluga, and examine this stock in some detail. This is done to illustrate the influence of harvest data, number of surveys, fit to the model, on our understanding of stock status. Following this two other stocks are discussed: CS beluga and EC-WG bowhead to show how the criteria might be used to decide what management framework could be applied. These other examples are included because the committee has recently dealt with them, is familiar with their status (Appendix 1) (Doniol-Valcroze et al. 2016; Marcoux and Hammill 2016), and they illustrate a range of information levels with respect to our understanding of cetacean stocks in Canada. However, additional input from stakeholders on long-term objectives and how those objectives might be achieved is still needed. The impacts of different approaches can also be examined further through more detailed simulation studies, within the framework of a management strategy evaluation (MSE).

MATERIALS AND METHODS

CRITERIA TO EVALUATE LEVEL OF STOCK KNOWLEDGE

With enough assumptions, a model can be fit to almost any data. However a population model fitted to the abundance estimates must be informative and provide a reasonable portrayal of the dynamics of the population rather than be driven by the assumptions. Some diagnostic parameters could include evaluating the goodness of fit of the model to the data and whether the model is providing useful insight. Are there additional independent data that might provide insights? Guidelines can include:

1. Certainty in stock composition/identification. Are there data to support stock delineation, stock composition of the harvest? Are harvest composition data incorporated into the model as a fixed (deterministic) value because there are no data or included as a sampling distribution (probabilistic)?
2. Are appropriate methods used to estimate abundance? Is there a time series of three or more abundance estimates available from the last 15 years, with the last estimate ≤5 years old? Are all estimates considered 'good' or were concerns raised during peer review? Are the estimates reasonably precise (e.g. CV < 30%)? Are the surveys separated in time sufficiently to provide an indication of trend?
3. Are there reliable harvest statistics? Are the data obtained from independent observers? Is there verification and what is the frequency of reporting (weekly, monthly, end of season)? Are data missing/frequently missing/rarely missing?
4. Are there other data that could provide insights into stock dynamics or trend (e.g. levels of mortality, reproduction, trends in mean age/sex composition of the harvest)?
5. What type of population model can be fitted to the abundance data (e.g. surplus production, age-structured)? Is there a reasonable estimate of historical abundance?
6. Does the model provide a reasonable fit? Does visual inspection of abundance estimates and model behavior appear reasonable?
7. Is the model robust to the assumptions that have been used?
8. Do model diagnostics suggest internal consistency with the data (e.g. are there signs of autocorrelation, convergence, cross-correlation)?

MANAGEMENT FRAMEWORKS

MSY framework

As outlined above, the LRL and the PRL are set as proportions of MSY, or in some cases, if MSY is not known or cannot be determined, these are set as proportions of Biomass at pristine levels (B_0). In places like New Zealand and Australia, fisheries are relatively young, and it is easy to estimate B_0 (e.g. Ministry of Fisheries 2008). In marine mammals, MSY is thought to occur around 60% of carrying capacity (K) (Range 50%-85%; Taylor and DeMaster 1993), which using the DFO approach results in a LRL at 24% of K and a PRL at 48% of K.

Atlantic Seal Management framework (ASM)

Because of the difficulties in estimating K, the Atlantic Seal Management Strategy uses the highest population observed or estimated (N_{max}) to set reference levels, where the PRL and LRL are set as 70% and 30% respectively of N_{max}. The management objective has been to maintain an 80% probability that the population will remain above the PRL.

Potential Biological Removal (PBR)

PBR was described above. It is estimated as:

$$PBR = N_{min} \cdot 0.5 \cdot R_{max} \cdot F_R$$

Where: N_{min} is the minimum estimated population size (usually calculated as the 20-percentile of the log-normal distribution around the estimate of N); R_{max} is the maximum rate of population increase with a default value for cetaceans of 0.04; F_R is a recovery factor (between 0.1 and 1),

(Wade 1998). The F_R that is applied depends on our understanding of stock status. Some guidelines for setting recovery factors are discussed later.

PBR is an estimate of total removals from the population that includes harvested animals, animals killed and not recovered, non-reported harvests and other types of human-induced mortality. The Total Allowable Take (TAT) is therefore:

TAT = PBR - (animals killed but not recovered + non-reported harvests + other anthropomorphic mortality). Other mortality includes bycatch, and ship-strikes.

MODEL SPECIFICATION

To examine possible frameworks, the basic stochastic stock-production model used to describe the dynamics of EHB, CS beluga, and Foxe Basin and Hudson Strait walrus was used (Marcoux and Hammill 2016; Hammill et al. 2016, 2017). Briefly, it is a state-space model that separates uncertainty associated with the observation process from the underlying population dynamics. It does not use sex or age structures, and incorporates density-dependence using a theta-logistic equation, with parameters λ_{max} (maximum growth rate), K (carrying capacity) and theta (which determines the shape of the density-dependence relationship). For the initial runs, λ_{max} was set at 0.04 and theta at 1, while K was estimated, A slightly different approach was used in the Nunavik assessment, where, λ_{max} , theta and K were estimated by the model (Hammill et al. 2017). The model is fitted to observations using MCMC in a Bayesian framework.

Hunters in Nunavik and Sanikilluaq (Nunavut) harvest animals belonging to an eastern Hudson Bay stock (EHB) and a western Hudson Bay stock (de March and Postma 2003; Turgeon et al. 2012; Doniol-Valcroze et al. 2016; Mosnier et al. 2017). To reflect the uncertainty in harvest composition, the proportions of EHB animals in the catch are incorporated in the model as statistical distributions. The raw catch data are included in the model separately for each harvest region (divided by season for 2009 and later). The resulting contribution of animals from the EHB stock to the overall harvest is then estimated within the model (Hammill et al. 2017).

The suitability of the model to explain the dynamics of EHB beluga was examined by fitting the model to 2,3,4,5 and 6 survey estimates in a series of separate exercises. In the first simulation series, the model start point was 1985, when the first aerial survey was completed. In the second simulation series, additional harvest information was included which allowed modelling of the population to begin in 1974. In a third approach, we examined the impact of including a longer series of catch data in the model; the historical catch data extending back to 1854 was included, and the model was fitted to the full time series of abundance estimates. Model diagnostics included checking for convergence, autocorrelation, cross-correlation, comparing model cv estimates to cv estimates from aerial surveys, and examining the probability that the population could increase over a period of 10 years in the absence of hunting.

Abundance information was available from six aerial surveys flown between 1985 and 2011 (Table 2). The EHB stock also has an extensive sampling and reporting network that has been in place since the mid-1990s, with weekly harvest data in Nunavik collected by independent monitors, who also collect information on other marine mammal observations and report unusual events in their weekly report. Hunters provide skin samples and a tooth from their catch. This information is used to provide information on stock identification (Mosnier et al. 2017), sex and age structure of the catch. Skin and tooth samples are also obtained from the subsistence hunt in Sanikiluaq (Nunavut). It should be noted, that these simulations are for illustration only since they were completed before the most recent abundance and stock composition data used for the stock assessment were available.

Table 1. Estimates of eastern Hudson Bay (EHB) obtained from aerial surveys flown between 1985 and 2011.

Year	Estimate	SE
1985	4282	557
1993	2729	1092
2001	2924	1404
2004	4274	1581
2008	2646	1244
2011	3351	1642

RESULTS

In the first set of runs, the 1985-2016 harvest time series was included and the model was fitted to different combinations of aerial survey estimates (Table 2). Model diagnostics showed rapid convergence, and no evidence of autocorrelation (indicating sufficient thinning of the MCMC chains). Some cross correlation was observed between variables. In cases with only a few survey points, there was correlation between the initial population size and the population size in 2016. When more aerial survey points were included, this correlation declined, but there was increasing correlation between carrying capacity (K), the initial population size (-ve) and between K and the current population size (+ve). This was offset to some extent by an increase in struck and lost.

The uncertainty around survey estimates of abundance are high (as is usually the case for beluga surveys), but as the number of aerial survey estimates available to the model increased, there was a decline in uncertainty associated with model trends as shown by a narrowing of the 95% credibility intervals. With four or more surveys, there was less uncertainty associated with the model estimate of abundance compared to the survey estimate of abundance (Table 2) (Fig. 2). However, some cross-correlation was observed between K and the most recent estimate of abundance (N2014 in figure) (Fig. 3).

Table. 2. Results from model runs including 1985-2016 harvest data and fitting to different combinations of aerial surveys assuming lambda=0.04. Change in estimates (95% credibility intervals) of carrying capacity (K), starting population in 1985 (Start), the population in 2016 (Pop 2016), probability of population increasing over a 10 year period if there is no hunting, and the coefficient of variation (cv) around the estimated 2016 population estimate. Numbers differ from the previous assessment due to differences in stock assignment values from the harvest. Letters refer to the panels in Figure2. #survey refer to the number of aerial surveys, values in parentheses (e.g. 85-01-11) refer to surveys flown in 1985, 2001, and 2011.

Variable\ #surveys	2	3b (85-01-11)	3c (85-93-01)	3d (01-04-08)	4	5	6
K ($x10^2$)	56 (33-234)	50 (34-233)	47 (31-235)	52 (34-187)	52 (35-238)	47 (34-235)	47 (35-234)
Start ($x10^2$)	44 (29-83)	42 (29-64)	39 (27-60)	43 (23-69)	41 (30-60)	40 (29-56)	40 (29-54)
Pop 2016 ($x10^2$)	39 (16-97)	35 (18-67)	32 (11-73)	36 (19-68)	37 (20-71)	33 (20-57)	33 (22-51)
Prob increase	0.6	0.7	0.65	0.7	.68	.77	.82
cv	.48	.35	.46	.33	.33	.27	.18

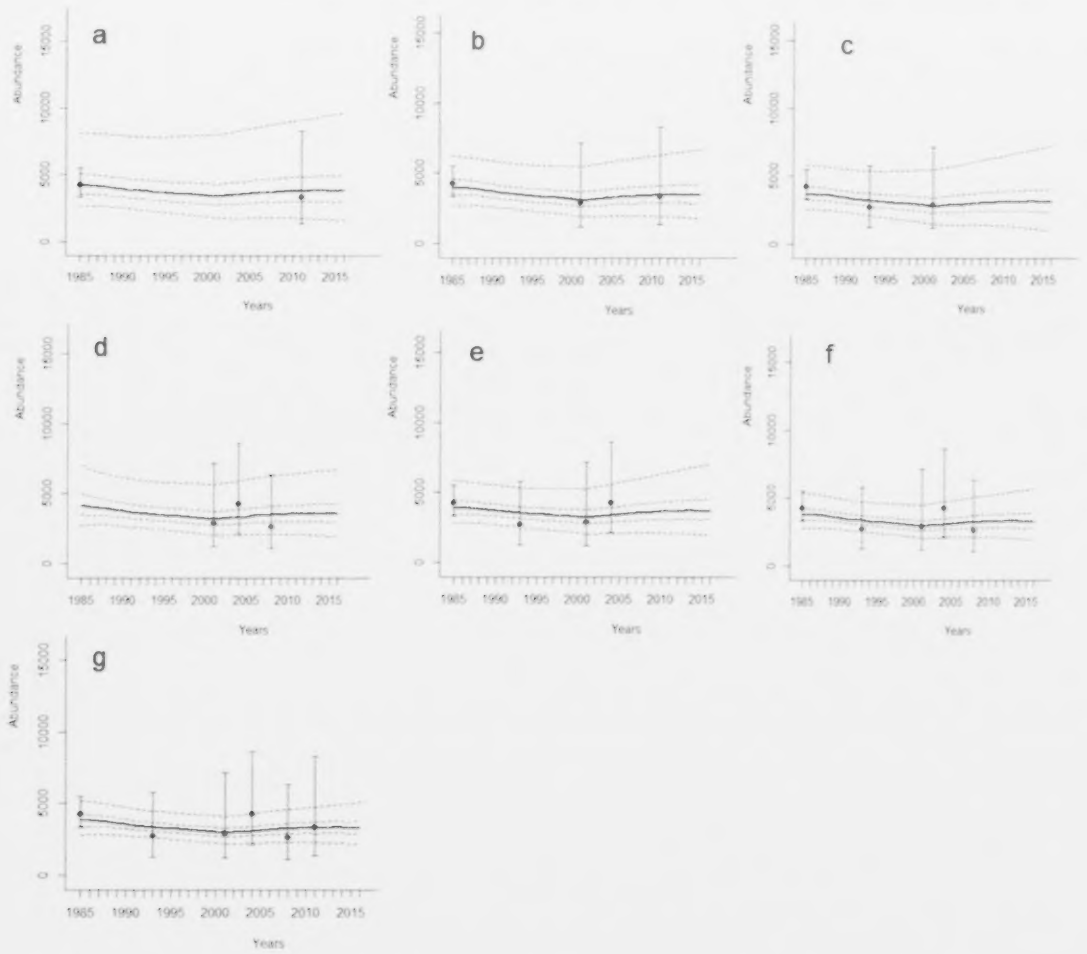

Figure 2. Aerial survey estimates (±95% CL) of abundance of EHB beluga from surveys flown between 1985 and 2011. The model was fitted to different number of surveys to obtain an estimate of abundance and trend. The median is the solid line. The inner dashed lines are the 25th and 75th quantiles, and the outer lines are the 95% CI. Letters refer to number of surveys: two surveys (a), three surveys (b,c,d), four surveys (e), five surveys (f) and six surveys (g).

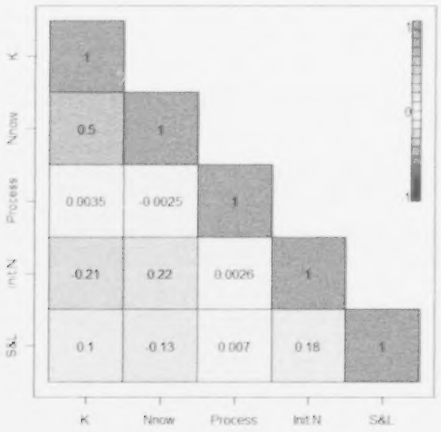

Figure 3. Cross-correlation among model parameters fitted to six aerial surveys and including harvest data from 1985-2016.

Additional harvest information (1974-1984) is available, and has been presented to the committee (Lesage et al. 2009). The overall impact of including the additional harvest data was a slight increase in the 2016 population estimate, a slight increase in the coefficient of variation around the 2016 estimate of abundance and increase in the probability of population growth if there was no hunting (Table 3). There was also a reduction in cross-correlation among model parameters (Fig. 4).

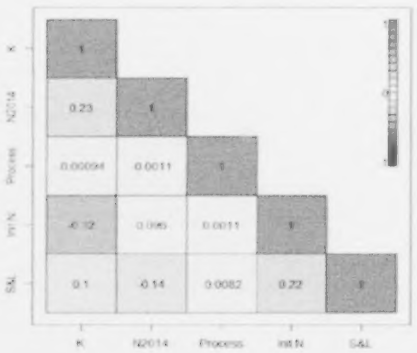

Figure 4. Cross-correlation among model parameters fitted to six aerial surveys and including harvest data from 1974-2016.

Reeves and Mitchell (1987) compiled harvest information from eastern Hudson Bay for the years 1854-1863. However, there are no data available for the intervening years between 1864 and 1973. Doniol-Valcroze et al. (2012) provided an estimate of pristine stock size assuming that at the end of the commercial whaling era there were a minimum of 1000 animals remaining in the population. Using a similar model to what is used here, they derived a pristine estimate of 8,000 belugas (95% CI 7,200-8,700), if there were no losses, and an estimate of 11,600 (95% CI=10,400-12,500), if hunting losses were as high as 50%.

Three additional runs were made, extending the assessment model back to 1854 and making different assumptions about possible harvests between 1864 and 1973. In the first run, the 1864-1973 catch was assumed to equal zero. In a second run, the 1869-1973 catch was assumed to equal the average of the 1974-2016 harvest from the eastern Hudson Bay arc. In a third run, the 1869-1973 harvest was assumed to equal the average of the 1974-2016 harvest for the Arc and the Hudson Strait-Ungava Bay (HSUB) region (Appendix 1, Table 1). Initial runs identified autocorrelation among parameters. To reduce this, the model was re-parameterized, setting the starting population equal to the carrying capacity (K), and allowing the model to estimate the maximum rate of increase and theta, the shaping parameter of the density dependent relationship.

Increasing the assumed number of animals taken between 1869 and 1973 increased the starting population, and suggested that the current trend of the population is a declining one (Fig. 5). Compared to the model runs starting in 1974 and in 1985, even assuming no catches between 1864 and 1973, the 1854 model run had lower probabilities that the population would increase and slightly higher coefficients of variation around the 2016 abundance estimate (Table 3). In all runs, the model showed cross correlation between lambda and K (-0.54 to -0.58).

Table 3. Impacts of including different harvest data in the model which also resulted in varying model start dates. All runs completed by fitting to 6 aerial survey estimates. Change in estimates (95% credibility intervals) of carrying capacity (K), starting population in 1985/1974 (Start), the population in 2016 (Pop_{2016}), probability of population increasing over a 10 year period if there is no hunting, and the coefficient of variation (cv) around the 2016 population estimate. Numbers may differ from the previous assessment due to differences in stock assignment values from the harvest.

Variable\ #surveys	1985 start	1974 start	1854 -no harvest data 1869-1973.	1854 - average 1974-2016 Arc harvest data for 1869-1973	1854 average 1974-2016 Arc and Strait harvest data for 1869-1973
K (x100)	47 (35-234)	72 (41-190)	103 (59-158)	138 (76-220)	159 (88-254)
Start (x100)	40 (29-54)	60 (28-11.2)	-	-	-
Pop_{2016} (x100)	33 (22-51)	34 (18-54)	29 (17-47)	29 (17-46)	29 (17-47)
cv	.18	.28	.25	.25	.25

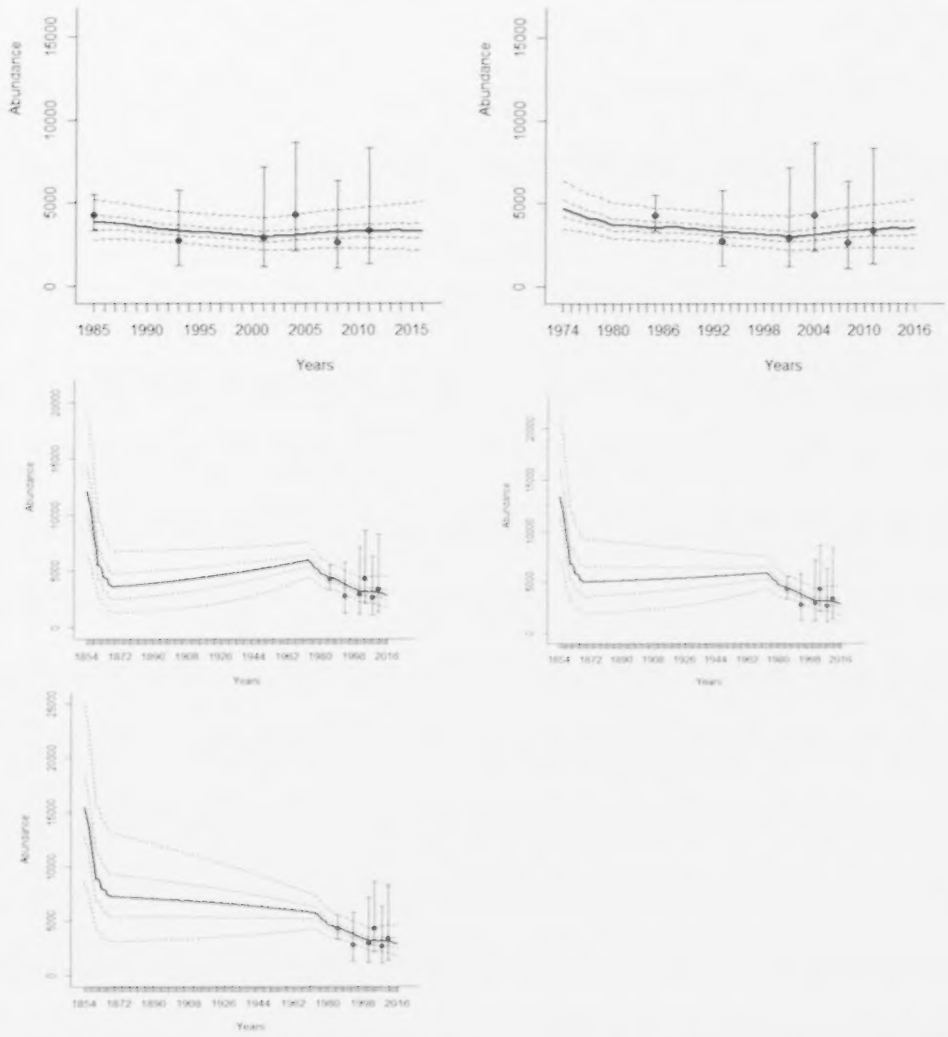

Figure 5. Trends of EHB belugas when different start dates and harvest data are included in the model fitted to aerial survey data (1985-2011). Model runs included 1985-2016 harvest data (top left), 1974-2016 harvest data (top right), 1854-1869 and 1985-2016, assuming no harvests during interim (middle left), 1854-1868 and 1974-2016 using average harvests (1974-2016) from Arc area during interim; 1854-1868 and 1974-2016 using average harvests (1974-2016) for Arc and Hudson Strait area during interim, taking into account stock composition of Hudson Strait harvest (bottom left).

DISCUSSION

Under a co-management regime, the Inuit right to harvest marine mammals is recognized and protected, but the land claim also calls for a management framework that respects the principles of conservation and protects the renewable resource economy. The PA provides a management framework that respects these objectives. A PA identifies regions where stocks can be considered healthy, cautious or in a critical state, and where appropriate limits on harvesting can be applied. As information on a stock improves, and depending on where it lies within the critical-healthy continuum, greater risks can be taken when establishing harvest limits.

A management framework has been developed for Atlantic seals that outlines conditions under which it is possible to identify healthy, cautious and critical zones and to identify explicitly the probability that harvests will allow the stock to recover or remain in the healthy zone. In situations, where the stock has not met the criteria, then it has been proposed that allowable removals be established using PBR. However, criteria are lacking for cetaceans and subsistence harvests in general. Marine mammals in the north have been the subject of considerable study, but the assessment data differ from that used to evaluate Atlantic seals. Guidelines that might be useful in deciding what management framework could be applied to cetaceans were identified earlier and are presented again here:

1. What is the certainty in stock composition/identification. Are there data to support stock delineation, stock composition of the harvest? Are harvest composition data incorporated into the model as a fixed (deterministic) value because there are no data or included as a sampling distribution (probabilistic)?
2. What time series of abundance estimates is available? For example for Atlantic seals a criterion is three or more abundance estimates available from the last 15 years, with the last estimate ≤5 years old? Are all estimates considered 'good' or were concerns raised during peer review? Are the estimates reasonably precise (e.g. CV < 30%)? Are different methods and approaches used to assess abundance? Is the entire stock surveyed or does the survey target a portion of the stock (e.g., age group)?
3. Are there reliable harvest statistics? Are the data obtained from independent observers? Is there verification and what is the frequency of reporting (weekly, monthly, end of season)? Are data missing/ frequently missing/rarely missing?
4. Are there other data that could provide insights into stock dynamics or trend (e.g. levels of mortality, reproduction, trends in mean age/sex composition of the harvest)?
5. What type of population model can be fitted to the abundance data (e.g. surplus production, age-structured)? Is there a reasonable estimate of historical abundance?
6. Does the model provide a reasonable fit? Does visual inspection of abundance estimates and model behavior appear reasonable?
7. Is the model robust to the assumptions that have been used?
8. Do model diagnostics suggest internal consistency with the data (e.g. are there signs of autocorrelation, convergence, cross-correlation)?

EHB, CS beluga and EC-WG bowhead were examined using these criteria. Following this brief analysis, the type of framework that could be applied to each stock is discussed. The examples used here are for illustration and discussion only.

EHB BELUGA

The EHB stock can be considered within the context of our criteria (Table 4). Hunters in Nunavik and Sanikilluaq (Nunavut) harvest from multiple beluga stocks. A sampling network has been collecting tissue samples and harvest data since the mid-1990s. In Nunavik, weekly reports are checked and unusual data are followed up with community wardens. In addition to the tissue samples, beluga teeth are also provided by hunters, which allow for monitoring age structure of the harvest over time (not reviewed here). As a result, there are reasonable data on the stock structure of the harvest and these harvest composition data are incorporated into a population model as statistical distributions (Points 1, 7 and 8). There is some uncertainty as to whether the genetic composition of animals taken in the harvest is representative of the genetic composition at large. Errors in this assumption will underestimate the impact of harvesting on the EHB

population (Point 1). Seven visual systematic line-transect aerial surveys have been flown, to estimate total abundance, four of them within the last 15 years (the 2015 survey not used in this analysis). The surveys are relatively consistent for the time-series (Point 2). The surplus-production population model that is fitted to the survey data takes into account reported harvests, appears robust, is internally consistent, and appears to provide a reasonable fit to the data (Points 2-8). Beginning the time-series in 1974 provides a better understanding of the dynamics than a model beginning in 1985 (Points 3, 4). The model shows that the population is currently stable or is increasing slightly, which is consistent with the management objective to set the Total Allowable Take (TAT) assuming that the probability of a population decline would not exceed 50% (Points 3-7). Extending the series back to 1974, by including additional harvest data increased the model estimate of K. Both models that started in 1985 and in 1974 converged rapidly and did not show signs of autocorrelation, but there was some cross-correlation between some variables (Points 4 and 5). Although there are some issues, we feel that the model beginning in 1974 provides a reasonable description of the dynamics of this stock.

Extending the time series back to 1854 provided some estimates for a historical value of K. However, harvest data are missing for almost 100 years (1868 to 1974). Different assumptions for subsistence harvests altered our understanding of historical population size and trend, as well as our understanding of current population trend, suggesting a current population decline, which is not supported by the recent survey estimates (Points 4-7). Also, K may have shifted since the 19th century, making its use in the current modeling inappropriate.

CUMBERLAND SOUND

The Cumberland Sound beluga population is harvested by hunters from Pangnirtung. Hunters have identified that there may be three types of beluga (Kilabuk 1998), although harvest information, along with biological samples have been unable to confirm local observations. Surveys are flown during August, when all belugas that are harvested by Pangnirtung in May-June are believed to be into the northern portion of Cumberland Sound. For management purposes hunters are assumed to be harvesting from a single beluga stock (Point 1). This stock has been of considerable interest since the 1970s, but survey coverage has changed over time. Surveys flown prior to 1990 focused on the Clearwater Fiord area where the majority of whales from Cumberland Sound occur in August, whereas surveys flown in 1990 and since then have extended their coverage to include areas outside of of Clearwater Fiord (Marcoux and Hammill 2016). Four surveys are available to evaluate model trend, but only two have been flown within the most recent 15 years (Point 2). A number of the survey estimates have been questioned (1990, 1999, and 2009). Problems with aircraft drift, and coverage occurred in 1990, camera problems and fog were encountered in 2009 (Richard 2013). The 1999 survey estimate is unusually high, but there does not appear to be any problems with this estimate which in fact represents the average of two surveys flown on nearly consecutive days with wide coverage.

Harvest information is available for this stock (Point 3). A surplus production model was fitted to the survey data and took into account information on harvests (Marcoux and Hammill 2016). The initial time-series included harvest data extending back to 1920, but problems were encountered with the model fit, so the time series was shortened to start in 1960. There is considerable uncertainty associated with the model fit (Points 5-8) to the CS beluga data (Fig. 6). Using different assumptions, all model runs indicate that the population is low, probably around 1,000 animals and is likely declining. However, additional survey and improved harvest information are needed for this stock (Marcoux and Hammill 2016) in order to model the population dynamics. There may be other demographic or harvest data that may improve the model, but these were not available for the assessment (Point 8) (Table 4).

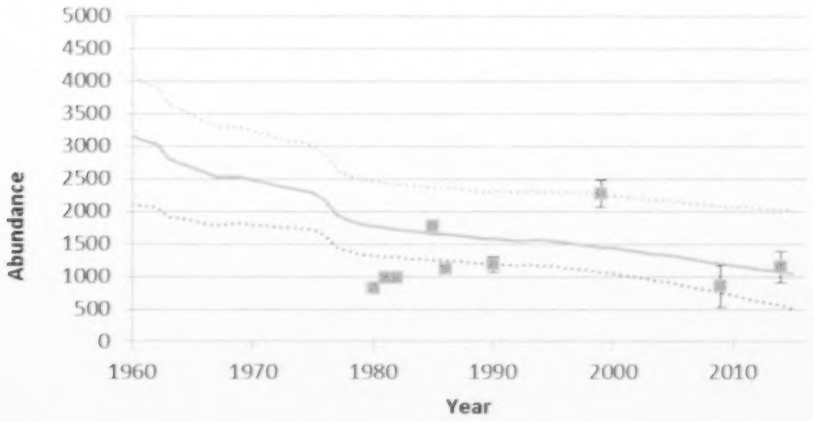

Figure 6. Model estimates of Cumberland Sound beluga abundance from preferred model fitted to aerial survey estimates flown in 1990, 1999, 2009 and 2014 corrected for animals at the surface (red squares with ± SE), and assuming θ of 1. Solid line shows the median estimates and dashed lines show 95% Credibility Intervals. Earlier surveys that only covered Clearwater Fiord 1980–1986 (red squares with no error bars) were not used for model fitting.

EC-WG BOWHEAD

The EC-WG bowhead stock was recently reviewed and there has been some exploratory modelling to determine historical population size (Higdon and Ferguson 2016; DFO 2015 Doniol-Valcroze et al. 2015). Up until the early 2000's this stock was considered to be two stocks, but telemetry and genetic information confirm the presence of a single stock in the eastern Arctic (Point 1). Animals from this stock are harvested by hunters from west Greenland and eastern Canada. There are 3 potential surveys that can be used for modelling, but only two have been completed in the last 15 years (Point 2). There is also a mark-recapture estimate available for 2013, but the aerial survey estimate from 2013 has been accepted as the official abundance estimate for the moment (DFO 2015) (Point 4). A survey of overwintering bowhead in Hudson Strait was flown in March 1981. Additional observations were available from the west Greenland coast, resulting in a combined estimate of 1,549 (95% CI 589-4,072) animals (Koski et al. 2006). The other surveys have been flown during summer; it is not clear if the winter and summer surveys can be compared since they do not cover the same areas and detectability of animals in the ice vs open water season in summer may differ (Point 2, 4). Surveys flown in 2002 did not cover the entire summer range of bowhead. Furthermore, these surveys have been subject to several different analyses resulting in substantially different estimates of abundance for the same data (Range-6,300-14,400) (Higdon and Ferguson 2016). The High Arctic Cetacean Survey, carried out in 2013, was designed to cover the entire Canadian summering range of EC-WG bowhead, but coverage of Foxe Basin was limited due to fog so this estimate is likely negatively biased as well. Thus, there is considerable uncertainty with respect to current abundance (Point 2). A long time series of harvest information is available extending back to the 1500s (Higdon and Ferguson 2016) (Point 3). Harvesting was prohibited after the 1920s. Harvesting re-started in the late 1900's and is closely monitored.

Some preliminary modelling explored possible pre-commercial population size, but the starting population calculations (15,000-20,000) reflects the range of model inputs that were explored (Higdon and Ferguson 2016). A surplus production model similar to that used for EHB beluga was fitted to the available harvest and abundance data (Appendix 2), but overall model fit to the

limited survey data was poor. Under this model, there is considerable uncertainty associated with the initial population size, current perceptions of the population, the form of the population decline and recovery and current population trend (Fig. 7) (Points 5-8). The median starting population is around 29,000 animals (95% CI=10,000-49,000) in either run, but the model fitted to two estimates of abundance, suggests that recovery was rapid, and that the current population has leveled off at 22,000 (95% CI=16,000-44,000) while in the case of fitting to three surveys, recovery has been more gradual with the population currently numbering around 13,000 (95% CI=5,000-44,000), but increasing rapidly (Fig. 7). Therefore, it appears that we have not identified an appropriate model that provides adequate fit to the data. In particular, it is likely that the 1981, 2002 and 2013 surveys did not cover the same proportion of the population, and are therefore not providing enough information for the model to capture the dynamics of this stock. Consequently, the modelled population trajectory and estimate of current population size is being driven by the assumptions for starting population, struck and loss, and the default R_{max} (Points 5-8).

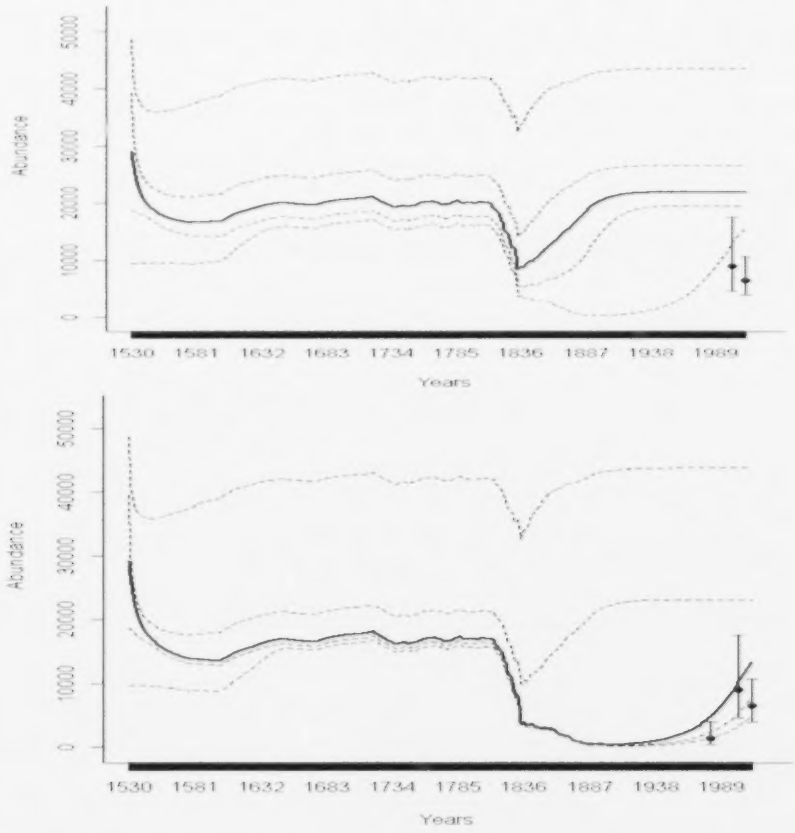

Figure 7. Estimated trends of EC-WG bowhead whales (1530-2013) obtained by fitting a population model to aerial surveys estimates (N=2 top, N=3 bottom) using Bayesian methods and taking into account historical catches. Median (solid line), outer dashed lines=95% CI, and 25th and 75th quantiles (inner dashed lines).

Table 4. Values applied to each criteria for each of the stocks evaluated in this study. For comparisons, different EHB beluga conditions were used (different start points, current vs historical data), the Cumberland Sound beluga and EC-WG bowhead whale.

Criteria	EHB 1974 start	EHB 1985 start	EHB 1854 start, no interim harvest	Cumberland Sound beluga	EC-WG Bowhead
1	Yes			1Yes	1Yes
2	Yes	Yes	Yes	No (17 y), problems	No, problems
3	Surplus production	Surplus production	Surplus production	Surplus production	No
4	Yes	Yes	Not pre-1974	Reasonable historic data, modern data uncertain	Good
5					
6	Yes	Yes some	No	No	No
7	Yes	Yes	No	Poor	-
8	Yes	Yes	No	yes-maybe	No

DATA RICH OR DATA POOR?

While stocks are often considered to be either Data Rich or Data Poor, this is really a continuum that represents our understanding of the dynamics of a stock or population. The term Data Rich simply refers to the situation where, based upon a review of a series of indicators of model performance and our knowledge of the stock, we feel that an appropriate model describes the dynamics adequately to provide management advice. If we do not have confidence in the population dynamics, management advice should be more cautious and the PBR approach could be used. This approach has undergone extensive simulations to ensure that it will not result in harm to the population (e.g. Wade 1998). Within the data poor category, however, there is also a continuum. For some species we have multiple surveys and some data that allows us to construct a model that, while not describing the full population dynamics, appears to provide an estimate of current abundance that is robust to the model assumptions. In other cases, there are insufficient data and model estimates are dependent upon the assumptions. As a result, the only reliable estimates we are left with are from the survey itself.

The three stocks we considered illustrate this range. According to our criteria, EHB beluga might represent a stock that could be considered as Data Rich, while, CS beluga and EC-WG bowhead could represent two different levels of Data Poor (Table 4). The abundance estimates for EC-WG bowhead are either dated or incomplete, and thus insufficient for model fitting. In this case, since there is only a single recent estimate of abundance, it is only possible to provide harvest advice using PBR based upon the survey estimate. In the case of CS beluga, there are more estimates of abundance, with three surveys flown within the last 15 years. However, overall, there are few surveys, and there have been some questions about some of the surveys. A population model has been fitted to the survey data and points to a declining population. While the model fit is poor, the estimate of current abundance is very similar under different sets of assumptions. This suggests that providing harvest advice using PBR is a more appropriate approach, but that using an estimate of current abundance from a model fitted to the survey data provides an approach to take into consideration the accumulated information for this stock.

RECOVERY FACTOR (F_R)

If PBR is to be used to provide harvest advice, then some discussion is also needed on the appropriate F_R to ensure consistency in its application. In the past, various recovery factors have been used for populations that appear to have similar levels of information and status. For example, F_R of 1 and 0.5 have been used for populations that have a COSEWIC designation of Special concern (or better), show no sign of decline, and are abundant (e.g. Foxe Basin, Penny Strait walrus; Stewart and Hamilton 2013; Hammill et al. 2016).

Based upon the status of a population and our understanding of the population trends, we propose criteria for the choice of F_R that are based on trends in abundance and the quality of available information. If a stock has a status that has been assigned by COSEWIC, then this status can also be used to provide guidance on what F_R to apply. These criteria are given in Table 5. A population that is abundant, and is increasing or stable would have an F_R of 1, while a stock which is abundant, but the number of abundance estimates are limited might have a F_R of 0.75, (F_R of 0.75 has not been used previously in Canada). A F_R of 0.5 could be used for a population that is abundant, but the stock is declining or trend is unknown. A stock that is small, and the trend is increasing or stable would have a F_R of 0.25. A population that is small and is declining or trend unknown would have a F_R of 0.1(e.g. St. Lawrence beluga, Cumberland Sound beluga), or perhaps a PBR should not be estimated, or assumed to be zero, if the stock is too small to allow removals.

Table 5. Proposed criteria for application of various levels of recovery factors (R_F) for use in Canada.

Level of recovery factors (R_F)	Population trend	Examples
1	Abundant, increasing or stable; COSEWIC designation: Not at Risk	Beaufort Sea beluga
0.75	Abundant, limited data, trend unknown but not considered to be declining; COSEWIC designation: Special Concern	WHB beluga
0.5	Abundant, declining or unknown if declining; COSEWIC designation: Threatened	High Arctic beluga
0.25	Small, increasing or stable; COSEWIC designation: Threatened or Endangered	EHB beluga
0.1	Small, declining or unknown, COSEWIC designation: Threatened or Endangered	Cumberland Sound beluga

PA FRAMEWORK FOR EHB BELUGA

In the case of EHB beluga, we have a reasonable level of understanding with respect to abundance and trend, and dynamics of the population. This indicates that a more structured framework could be applied to this stock, where the probability of different harvest levels allowing the population to meet management objectives related to LRL, PRL and Target levels could be estimated. Below we describe two possible frameworks.

International agreements have identified MSY as a management objective. In fisheries, the best estimates of MSY are obtained if stock biomass has varied over time and the time series covers a range of productivity conditions. Unfortunately, many marine mammals have been heavily exploited, and in many cases current populations are way below MSY; some have poor conservation status, although others have recovered (see Magera et al. 2013). To estimate MSY, information on ecosystem carrying capacity (K) and the shape of the density dependent relationship (theta) are needed. For many species, historic catch data have been used to reconstruct the pre-commercial hunt population size, which is assumed to have been at K. This approach may be valid if the uncertainty in the harvest data is understood and if environmental carrying capacity has not changed. In the case of EHB beluga, the historical catch estimates are based on quantities of oil and the trade in half skins, with assumptions being made about the average oil yield per whale, which must vary with animal size and reproductive status; the catch data reportedly represent catches from only two areas (Little Whale and Great Whale rivers) and do not contain information from subsistence harvests along the eastern Hudson Bay coast, nor in Hudson Strait. Furthermore there is a gap of over 100 years in the harvest reports. During that time, hunters moved from hunting from small camps using kayaks and harpoons to development of communities, increasing use of rifles and motorized vessels. Therefore it is difficult to model catches over this time. However, perhaps the most important factor has been a change in ecosystem conditions with declining sea levels in the Hudson-James Bay complex, shortening of the ice-covered season that will affect seasonal productivity and an apparent change in the timing of the spring and fall migrations of beluga. Construction of hydro-electric dams has altered the freshwater cycle within the Bay, (Tsuji et al. 2009; Galbraith and Larouche 2011; Hammill 2013), suggesting that an estimate of K from the late 1800s is not valid as an estimate of K under current conditions. An alternative to using a historic K might be to examine other proxies. If scientists, managers and stakeholders accept that they represent proxies, and build into the management framework a means of updating these indices as new information is obtained, then a framework that identifies Healthy, Cautious and Critical zones can be developed. This was also the reasoning behind the use of N_{max} as a reference point for the ASMS (Hammill and Stenson 2007).

To apply a more structured management framework it is necessary to identify levels for the LRL and PRL. Under the DFO MSY approach, the model estimate of K could act as a proxy in setting the PRL and LRL. To avoid confusing this 'K' with true ecosystem carrying capacity, we rename this parameter K_{model}. The model fitted to the 1985-2011 aerial survey data and including the 1974-2016 catch data produced an estimated K_{model} of 7,200 (Table 4). The PRL for this stock will be set at 48% of K_{model} (assuming maximum productivity occurs at 60% of K) and the LRL at 24% of K_{model}. For EHB beluga this will result in PRL and LRL of 3,500 and 1,700 respectively. The current estimated population (median) from these model runs is 3,400. The probability that the population is above the LRL is 1, while the probability that the population is above the PRL is only 0.45.

Using the Atlantic Seal Management approach, the PRL and LRL are set at 70% and 30% respectively of the largest population observed or the largest population estimated from the model (N_{MAX}). For EHB beluga, the largest population estimated by the model was 6,000

animals. Using N_{MAX}=6,000, results in LRL and PRL levels of 4,200 and 1800 respectively, which are very close to the levels identified using MSY.

Under both metrics the EHB stock is in the cautious zone. Within the cautious zone, the objective is to identify harvest levels that will allow the resource to move to the healthy zone within a set time frame.

For a stock in the Cautious zone, the type of harvest rules to be developed may depend on the proximity of the stock to the Critical zone, and whether the stock is declining, stable or increasing. Overall, for a stable or increasing stock (e.g. EHB beluga), the probability of a decline should be < 0.5 and management actions should promote stock growth to the Healthy zone, with a high probability (≥0.75), within a reasonable time frame, where a reasonable time frame may be considered 1.5 to 2 generations (39-52 years for beluga assuming 1 growth layer group per year; Stewart et al. 2006). ASM suggests that managers should aim for a probability that the population is above the LRL of 0.95 and that the probability that the population is above the PRL is 0.8 within about 1 generation. In New Zealand, stock rebuilding should occur within T_{HO} and $2*T_{HO}$, where T_{HO} is time in years with no harvest (Ministry of Fisheries 2008). Under the DFO system, a harvest of 20 EHB animals per year would allow the population to recover to the Healthy zone with a probability of 0.75 after 26 years ($2*T_{H0}$) (roughly 1 generation). A harvest of 30 animals will allow the population to move into the Healthy zone with a probability of 0.75 within 54 years (2 generations) (Fig. 8). However, this projection is likely to be conservative because it assumes increasing uncertainty as time since the last survey was flown increases. As uncertainty increases, a higher abundance estimate is needed for the population to exceed the threshold with a probability of 0.75. In reality, an abundance survey is likely to be flown every five years. Each time a new survey is flown, and the model refitted to the abundance estimates, the uncertainty associated with the estimate is reduced.

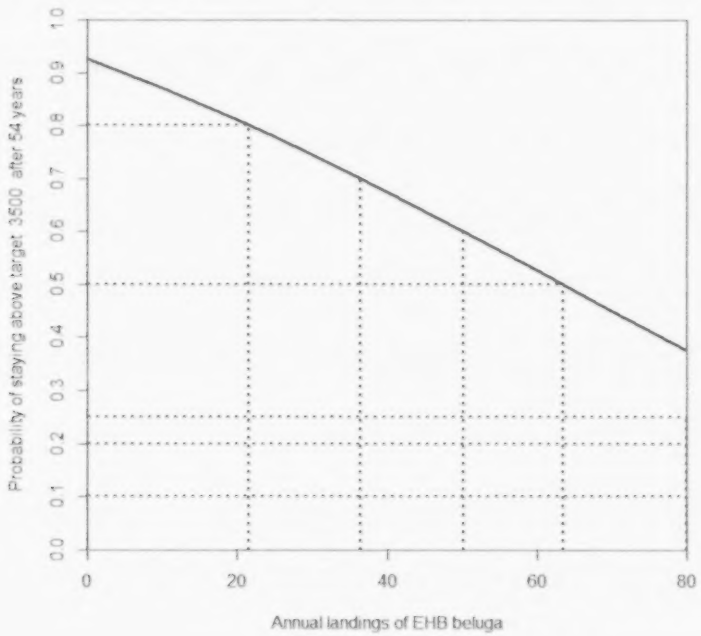

Figure 8. The probability of the EHB beluga population being above the PRL of 3,500 within 2 generations (54 years) under different harvest scenarios using the MSY framework.

Under the ASM approach, a harvest of 20 EHB animals per year would allow the population to recover to the Healthy zone with a probability of 0.8 after 26 years ($2*T_{H0}$) (roughly 1 generation). A harvest of 28 animals will allow the population to move into the Healthy zone with a probability of 0.8 within 54 years (2 generations), while a harvest of 37 EHB whales would allow the population to move into the Healthy zone within 54 years with a probability of 0.75 (Fig. 9).

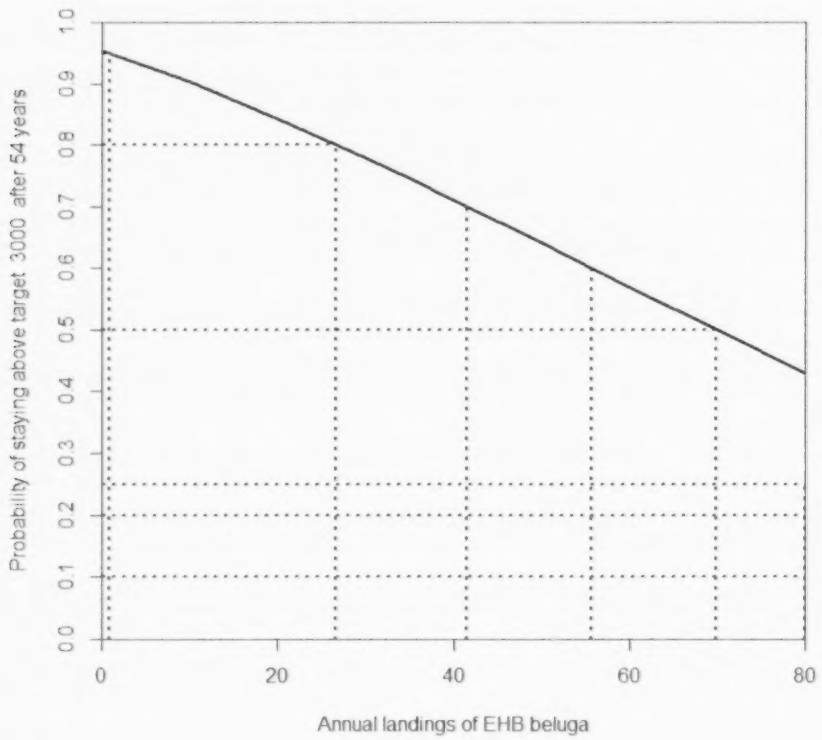

Figure 9. The probability of the EHB beluga population being above the PRL of 3,000 within 2 generations (54 years) under different harvest scenarios using the ASM framework .

In summary, the criteria identified above follow the general frameworks identified in the DFO policy (DFO 2006) and ASM (Hammill and Stenson 2007, Stenson et al. 2012). However, a review of the frameworks applied within the Department would show that there is considerable variability in approaches, between species, regions and stocks, which reflect our understanding of the stock status, its productivity regime and the types of monitoring data that are collected (DFO 2013). We have attempted to develop an approach that reflects the type of data available for cetaceans.

The objective of the PA is to manage populations around the PRL. For populations that are well studied a PRL and LRL can be specified and the TAT can be estimated along with the probability that a harvest will allow the resource to recover to or remain above the PRL. In cases where PRL and LRL cannot be determined, harvests could be estimated using PBR. Under PBR, the objective is that the resource will recover to a level above the OSP, or will remain above the OSP within 100 years with a probability of 0.95. To assist in evaluating where a stock may lie along the Data Rich-Data Poor continuum, several criteria were developed (and applied

to EC-WG bowhead, CS and EHB beluga as illustrations) to determine which PA framework might be most appropriate. For EHB beluga it is possible to identify explicitly the PRL and LRL and two approaches were outlined for how this might be done. For bowhead and CS beluga, it was not possible to identify the PRL explicitly, but different approaches in applying PBR were also outlined.

The Precautionary Approach provides a mechanism for scientists, managers and harvesters to identify clear management objectives that will minimize the risk of resource over-exploitation. At the same time, it is important that harvester's concerns are incorporated into the framework, to minimize the costs of lost harvest opportunities. In this sense this document represents a start to the process and additional input is needed from managers and stakeholders. Other aspects that could be discussed include adding additional criteria to evaluate our understanding of stock status and trend, adjusting the minimum number of surveys over a time frame to take into account species' generation time, defining what is abundant, or including additional criteria for the assignment of the Recovery Factors. If a framework can be established, it will allow Canada to respect both the land claim and domestic/international policy/agreements.

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APPENDIX

Appendix 1. Table 1. Reported harvests and zones animals were harvested from. The zones are the eastern Hudson Bay arc (ARC), Hudson Strait-Ungava Bay (HSUB), Sanikiluaq (SAN), Hudson Strait Spring (Spring) and Fall (FALL), Ungava Bay Spring (UBSP) and Fall (UBFA), Northeastern Hudson Bay spring (NEHBSP) and Fall (NEHBFA)

YEAR	ARC	HSUB	SAN	SPRING	FALL	UBSP	UBFA	NEHBSP	NEHBFA
1854	423	0	0	0	0	0	0	0	0
1855	707	0	0	0	0	0	0	0	0
1856	747	0	0	0	0	0	0	0	0
1857	1366	0	0	0	0	0	0	0	0
1858	1023	0	0	0	0	0	0	0	0
1859	1043	0	0	0	0	0	0	0	0
1860	1511	0	0	0	0	0	0	0	0
1861	30	0	0	0	0	0	0	0	0
1862	229	0	0	0	0	0	0	0	0
1863	796	0	0	0	0	0	0	0	0
1864	165	0	0	0	0	0	0	0	0
1865	136	0	0	0	0	0	0	0	0
1866	504	0	0	0	0	0	0	0	0
1867	139	0	0	0	0	0	0	0	0
1868	188	0	0	0	0	0	0	0	0
1974	119	352	0	0	0	0	0	0	0
1975	137	532	0	0	0	0	0	0	0
1976	143	403	0	0	0	0	0	0	0
1977	181	501	0	0	0	0	0	0	0
1978	120	174	0	0	0	0	0	0	0
1979	211	224	0	0	0	0	0	0	0
1980	220	212	0	0	0	0	0	0	0
1981	61	236	0	0	0	0	0	0	0
1982	73	271	0	0	0	0	0	0	0
1983	69	227	0	0	0	0	0	0	0
1984	97	189	0	0	0	0	0	0	0
1985	78	166	5	0	0	0	0	0	0
1986	43	126	25	0	0	0	0	0	0
1987	53	125	28	0	0	0	0	0	0
1988	52	117	20	0	0	0	0	0	0
1989	84	284	19	0	0	0	0	0	0
1990	53	109	20	0	0	0	0	0	0
1991	106	178	22	0	0	0	0	0	0
1992	78	96	20	0	0	0	0	0	0
1993	67	189	10	0	0	0	0	0	0
1994	82	207	50	0	0	0	0	0	0

YEAR	ARC	HSUB	SAN	SPRING	FALL	UBSP	UBFA	NEHBSP	NEHBFA
1995	55	221	30	0	0	0	0	0	0
1996	56	211	30	0	0	0	0	0	0
1997	51	239	19	0	0	0	0	0	0
1998	50	252	54	0	0	0	0	0	0
1999	57	238	32	0	0	0	0	0	0
2000	62	208	23	0	0	0	0	0	0
2001	73	241	27	0	0	66	0	0	0
2002	5	161	15	0	0	23	0	0	0
2003	8	168	80	0	0	26	0	0	0
2004	3	144	94	0	0	4	0	0	0
2005	1	172	53	0	0	5	0	0	0
2006	0	147	22	0	0	2	0	0	0
2007	21	165	24	0	0	6	0	0	0
2008	23	92	33	0	0	5	0	0	0
2009	21	0	34	68	70	6	0	0	0
2010	16	0	47	138	61	8	7	0	0
2011	19	0	32	115	86	0	17	0	0
2012	13	0	61	208	56	10	2	0	0
2013	8	0	76	150	90	8	0	0	0
2014	22	0	26	208	37	11	0	1	14
2015	36	0	170	106	94	28	3	0	30
2016	11	0	33	117	0	20	0	0	0

Appendix 2. EC-WG Bowhead

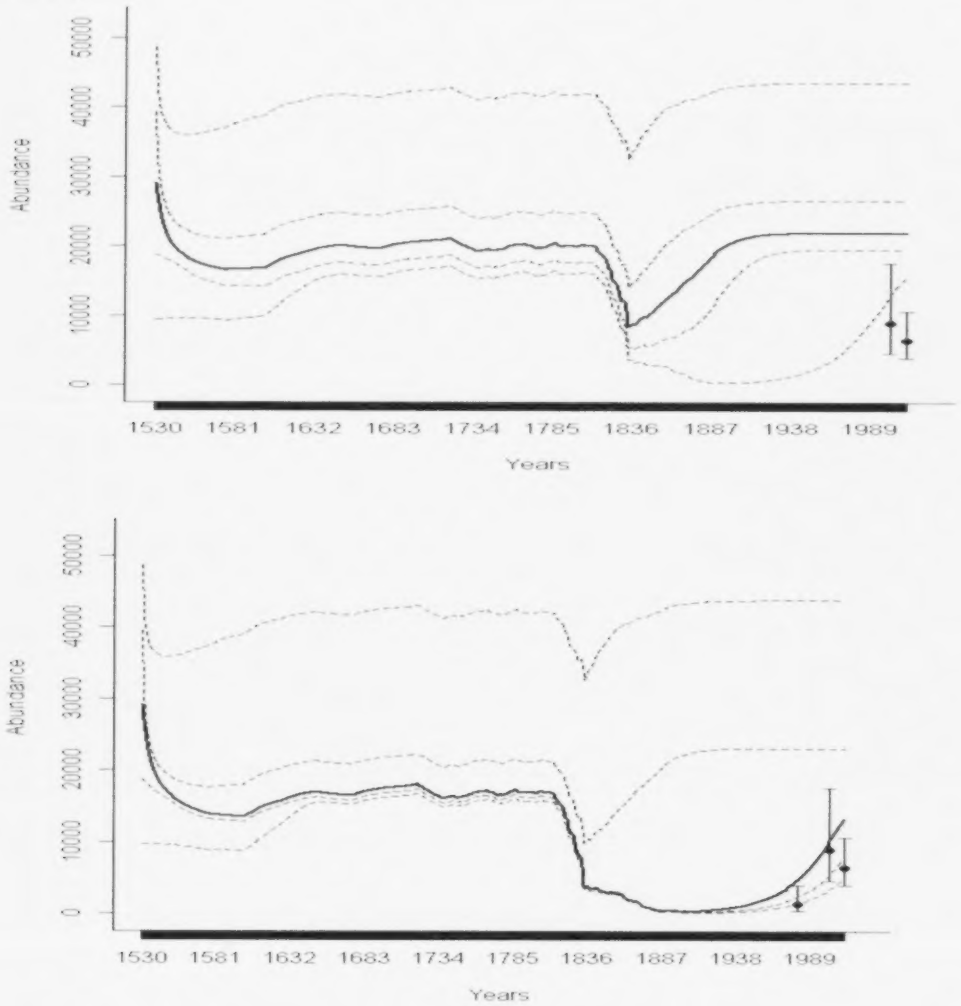
EC-WG Greenland bowhead, extensive catch data are available, but abundance data are limited to two surveys completed by DFO while a third estimate is available from winter surveys flown in 1981 (Higdon and Ferguson 2016). Monitoring of the bowhead hunt is extensive. DFO observers are present onsite to observe the hunt and collect biological samples.

In a recent exploratory analysis, Higdon and Ferguson (2016) used a deterministic approach to reconstruct the EC-WG bowhead population from historical catch data. They estimated the historical population using a model that assumed, as priors, that K=18,000-19,000, struck and loss varied between 10% and 20%, R_{max} was between 0.035 and 0.045 =, and the density dependent shaping parameter was initially set at 2.39. Their estimate of historical abundance was approximately 18,500 whales (95% CI=18,022-18,972), with R_{max} = 0.04 (SD=0.0029) and S&L=1.1506(95%CI-1.1027-1.1973).

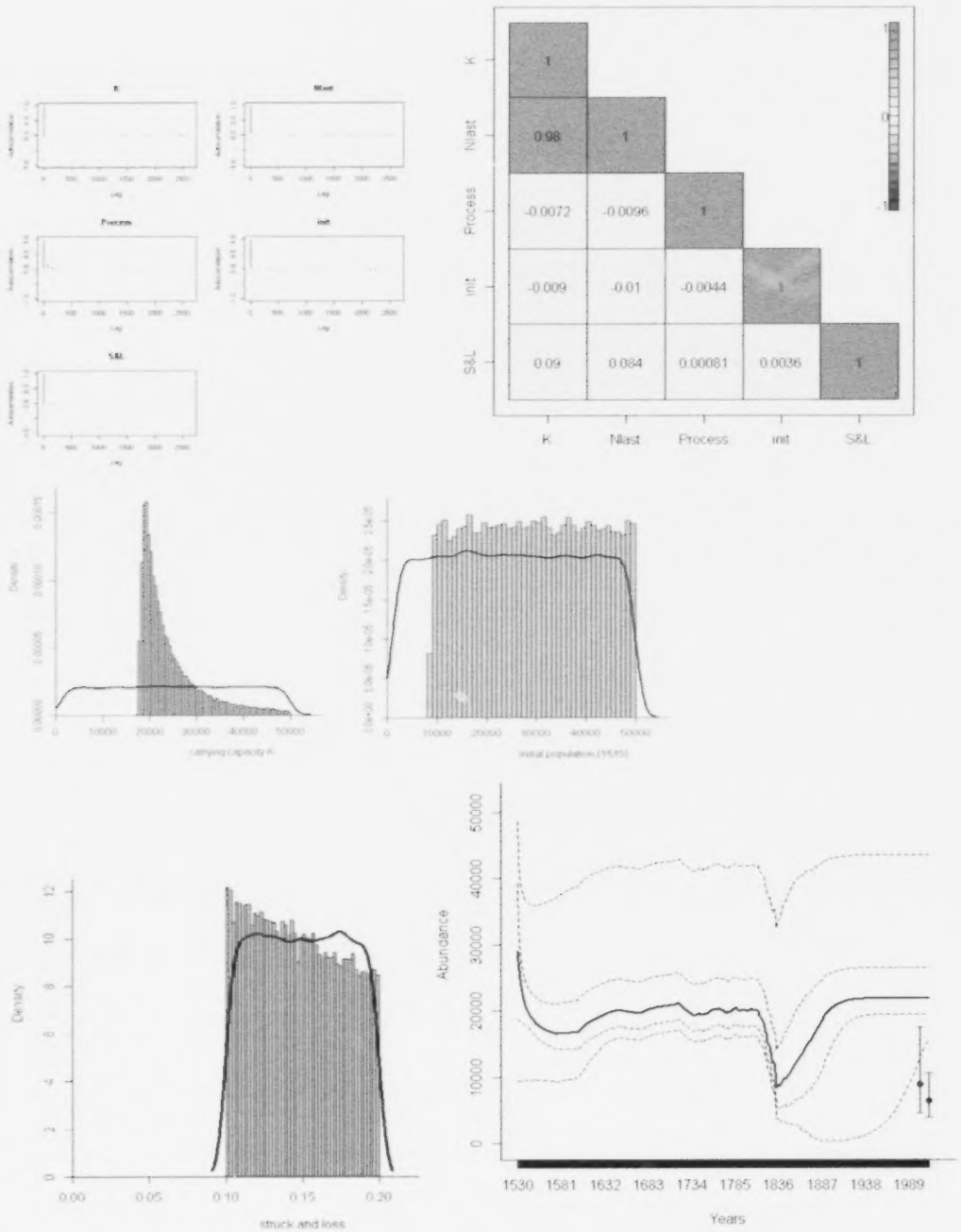
To provide some consistency in comparisons with the two beluga stocks, we fitted the 'beluga' model to the available bowhead abundance and catch information. State-space models, particularly when applied in a Bayesian framework, are a means of integrating data with population dynamics models and simultaneously quantifying the various types of uncertainty (Buckland et al. 2007), which makes this approach particularly suited for application to bowhead, where the data are limited and very uncertain. Values from Higdon and Ferguson (2016), were used as priors for the model.

When the 2002 survey was flown, EC-WG bowhead were considered to consist two separate stocks and consequently, the entire summer range was not surveyed in a single season. The 2002 survey has been re-analysed several times, resulting in a range of abundance estimates (6300-14,400) (Higdon and Ferguson 2016). The average from these analyses is 8948 (SE=3161). In 2013, a second summer aerial survey was flown, but due to weather, cover of Foxe Basin was not complete. This survey resulted in an estimate of 6,446 (SE=1484). A mark-recapture estimate (7,660, Frasier et al. 2015) is available for 2013, but the NMMPRC recommended that this estimate not be used until more information was available with respect to possible biases using this approach (DFO 2015). Combining two independent winter surveys, Kosiki et al (2006) estimated a population of 1,349 (SE=1600) animals in 1981 (Koski et al. 2006, Higdon and Ferguson 2016), but these surveys have not been reviewed within NMMPR. Making some assumptions about population growth rates, Koski et al (2006) projected this estimate forward to 2004 (3,633, 95% CL=1382-9550), but this estimate is not independent so cannot be used further here. Fitting the model to the two or three aerial survey estimates, and including harvest data extending back to the 1500s, resulted in different portrayals of the dynamics of this population (Fig. 6). There was a little difference in estimates of K, or starting population, between model runs with two surveys (K median=21,900, 95% CI=18,000-43,600, rounded to nearest 100; start population median=21.800, 95% CI=15,500-43,600) or three surveys (K median=18,900, 95% CI=17,600-43,800; start population median=29,500, 95% CI=10,000-49,000) (Appendix 2, Tables 1, 2). The two survey model showed a population that has leveled off at 21,800 (95% CI=15,500-43,600), while the three survey model showed an increasing population currently numbering 13,200 (95% CI=5,000-23,100). The population reached a minimum of 8,400 (95% CI=3,700-32,400) in 1834 for the two survey model, while for the three survey model, the population reached a minimum of 300 animals (95% CI=100-43,400) in 1905. The two survey model converged rapidly, but showed negative cross correlation between the estimated rate of increase (lambda) and the 2013 estimate of abundance (r=0.98). Some updating of priors was observed, (Appendix 2, Fig 1, Table 1). In the three survey model there was no evidence of autocorrelation, but there was strong cross-

correlation between K and the current (2013) estimate of abundance(r=0.91) (Appendix 2, Fig 2, Table 2).



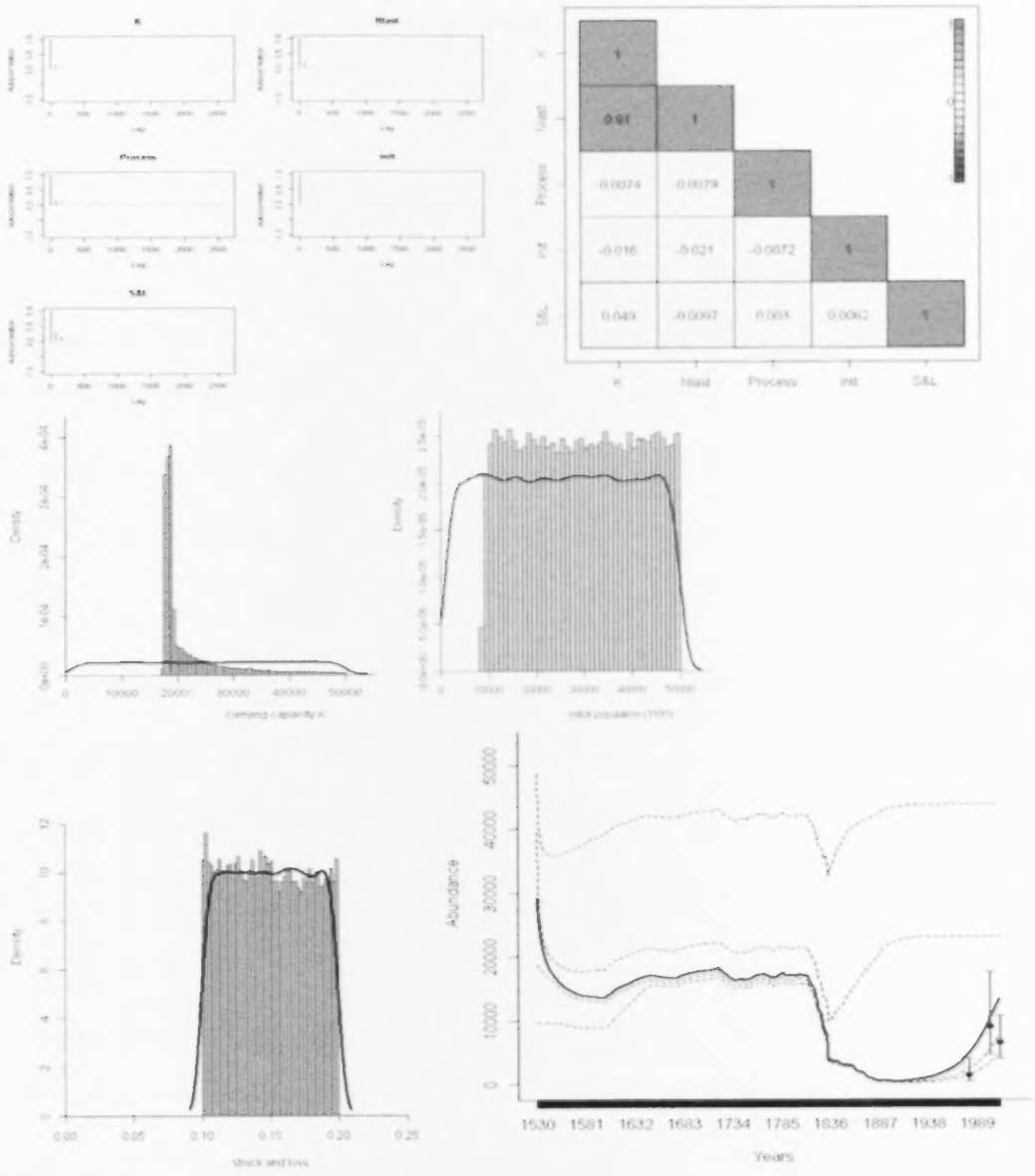
Appendix 2, Figure 1. Estimated trends of EC-WG bowhead whales (1530-2013) obtained by fitting a population model to aerial surveys estimates (N=2 top, N=3 bottom) using Bayesian methods and taking into account historical catches.



Appendix 2, Figure 2.Two survey model showing level of autocorrelation (top left), cross correlation (top right), prior and posterior distributions and population trajectory. Surveys (±95%CL), median (solid), 25th,75th quantile (inner dotted lines) and 95% CI (outer dotted lines). Theta fixed=2.39.

Appendix 2 Table 1. Model outputs for EC-WG bowhead stock using two surveys with 1530–2013 catch history. The mean, standard deviation (SD), 2.5th, 25th, 50th, 75th and 97.5th quantiles are given for the following model parameters and their priors: carrying capacity (K), process error (process), survey precision (survey), struck and lost (SL), and population size in 2013. $\hat{R}$ is the Brooks-Gelman-Rubin statistic; values near 1 indicate convergence of chains. N.eff is the number of effective runs after considering autocorrelation. Lambda=0.04, theta=2.39

Model parameters	Mean	SD	2.5%	25%	50%	75%	97.5%	Rhat	n.eff
K	24268	6625	17966	19557	21876	26594	43607	1.001	43000
K.prior	25542	14092	2271	13405	25501	37767	48769	1.001	50000
Deviance	43	2	40	42	43	44	49	1.001	50000
Prec.process	1480	1219	102	594	1164	2023	4643	1.001	46000
Prec.process.prior	1497	1211	108	610	1185	2058	4638	1.001	50000
Prec.surv	2.25	1.527	0.435	1.21	1.906	2.892	6.017	1.001	50000
Prec.surv.prior	6.256	3.963	1.053	3.339	5.445	8.276	16.017	1.001	27000
Startpop	29350	11898	9750	19107	29328	39646	48982	1.001	50000
Startpop.prior	25492	14105	2250	13367	25463	37673	48767	1.001	45000
SL	0.147	0.029	0.102	0.122	0.146	0.172	0.197	1.001	40000
SL.prior	0.15	0.029	0.102	0.125	0.15	0.175	0.197	1.001	50000
N2013	24038	6918	15515	19527	21847	26565	43578	1.001	50000



Appendix 2, Figure 3.Three survey model showing level of autocorrelation (top left), cross correlation (top right), prior and posterior distributions and population trajectory. Surveys (±95%CL), median (solid), 25th,75th quantile (inner dotted lines) and 95% CI (outer dotted lines). Theta fixed=2.39, Lambda fixed=0.04.

Appendix 2 Table 2. Model outputs for EC-WG bowhead stock using three surveys with 1530–2013 catch history. The mean, standard deviation (SD), 2.5th, 25th, 50th, 75th and 97.5th quantiles are given for the following model parameters and their priors: carrying capacity (K), process error (process), survey precision (survey), struck and lost (SL), and population size in 2013. $\hat{R}$ is the Brooks-Gelman-Rubin statistic; values near 1 indicate convergence of chains. N.eff is the number of effective runs after considering autocorrelation. Lambda=0.04, theta=2.39

Model parameters	Mean	SD	2.5%	25%	50%	75%	97.5%	Rhat	n.eff
K	22176	6893	17554	18208	18869	23105	43799	1.022	200
K.prior	25579	14129	2222	13290	25587	37849	48734	1.001	50000
deviance	60	5	54	55	59	64	69	1.008	410
prec.process	1507	1224	113	612	1195	2068	4672	1.001	22000
prec.process.prior	1503	1227	108	606	1188	2059	4665	1.001	50000
prec.surv	2.857	2.833	0.208	0.618	1.623	4.548	9.913	1.008	410
prec.surv.prior	6.275	3.972	1.026	3.369	5.461	8.292	16.135	1.001	50000
startpop	29462	11901	9974	19104	29472	39813	49005	1.001	50000
startpop.prior	25476	14151	2181	13170	25520	37717	48801	1.001	46000
SL	0.149	0.029	0.102	0.124	0.149	0.175	0.198	1.001	7100
SL.prior	0.15	0.029	0.102	0.125	0.15	0.175	0.197	1.001	50000
N2013	16939	10747	5017	7837	13214	23076	43771	1.011	290